LCA Methodology

Generic Spatial Classes for Human Health Impacts, Part I: Methodology

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Preamble. This series made up of two articles is devoted to a new method for the spatially differentiated assessment of the impacts of primary airborne pollutants on human health within Life Cycle Assessments. The first part describes the method and provides exemplary results for site-dependent exposure efficiencies. The second part deals with the application of the method within a Life Cycle Assessment of natural gas vehicles.

Abstract. A new method for the spatially differentiated assessment of impacts of airborne pollutants on human health is presented. It is applicable to primary pollutants with linear exposure response functions. This includes the most important primary air pollutants from transportation and energy generation. The article looks at the spatial differentiation of impacts due to emission height and the local population density distribution around the emission site, as has been predicted using a Gaussian plume model. The differentiation due to population density is captured by way of five generic spatial classes: large cities in agglomerations, highly densified districts in agglomerations, cities in urbanized regions, country average districts, and low density rural districts in rural regions. Average impacts are calculated for each class. The method is simple enough to be applied to a large number of emissions within Life Cycle Assessments. It was used to calculate site-dependent exposure efficiencies for a variety of primary pollutants emitted at different heights. For traffic emissions of pollutants with short atmospheric residence times, the exposure efficiencies vary by a factor of 5 across Germany and by a factor of 75 across Europe. This differentiation due to population density decreases significantly with an increasing atmospheric residence time of the pollutants and with an increasing emission height.

Keywords: Airborne pollutants; emission height; exposure efficiency; human toxicity; Life Cycle Impact Assessment (LCIA); population density; spatial differentiation

Introduction

Methods for the spatial differentiation of the assessment of local or regional impacts within Life Cycle Assessment (LCA) are a topic of current interest. This applies to human health impacts in particular. The present paper addresses this issue for the case of carcinogenic and respiratory health impacts of primary airborne pollutants. Only atmospheric exposures are taken into account. For the most important air pollutants from transportation and energy generation, a linear exposure-response function can be assumed (Spadaro and Rabl 1999). In this case, the marginal impact of an emission with a given mass depends upon the emission height, as well as on the population density distribution and the meteorological conditions (but not on the background concentration of the pollutant) around the emission source. In principle, it is desirable to take these spatially variable parameters into account in the impact assessment. However, due to the large number of processes to be considered in an LCA, there is a trade-off between the spatial detail and the ease of use of impact assessment methods.

Existing methods of impact assessment are located at different positions along this trade-off. The methods commonly used at present are generic, i.e. they do not include any spatial differentiation. Examples are the method of Critical Volumes (Habersatter 1990), the CML method (Heijungs et al. 1992) and the Critical Surface Time method (Jolliet and Crettaz 1997). Hofstetter (1998) considers variations of the population density at a sub-continental scale (i.e. between countries or sub-continental regions) in the context of multimedia models, but does not include differentiations with regard to emission height or settlement structures at a local scale, i.e. between cities and rural areas.

At the other end of the trade-off spectrum, the use of site-specific impact pathway analyses in the context of LCA has been proposed by Krewitt et al. (1998). Exemplary damage factors for a number of European sites are provided by Spadaro and Rabl (1999). A software tool is available to facilitate such site-specific assessments (IER 1998). However, site-specific meteorological data (including local wind directions) still need to be added by the user. Their collection might not always be feasible for a large number of sources or for traffic sources which pass by many different sites. Potting (2000) provides a framework for the calculation of site-specific impacts, which is based on (IER 1998). Typical meteorological data for four zones within Europe are used, but the issue of local wind direction is not addressed, and no operational guidance for the determination of local population densities is provided. Moriguchi and

Terazono (2000) also avoid the issue of local wind directions by hypothetically assuming a uniform distribution without justification.

The method presented here provides both a detailed justification for avoiding the tedious collection of site-specific wind directions and operational data for local population density distributions. It follows an intermediate way between a generic and a site-specific approach: The differentiation of the health impacts between cities and rural areas is approximately captured by a limited number of generic spatial classes, as suggested by Potting and Hauschild (1997) and recommended by Udo de Haes et al. (1999) (section 1). On the basis of generic spatial classes defined for Germany, site-dependent exposure efficiencies were calculated for a variety of airborne pollutants (section 2).

1 Methodology

Consider the emission of a mass M of a primary air pollutant with a linear exposure response function of slope E (effect factor) from a point source located at site i. Its incremental health impact ΔD_i aggregated across the entire population can be written as

$$\Delta D_i = E\, M\, \Delta I_i = E\, M\, EE_i / V \tag{1}$$

where ΔI_i is the incremental population exposure per emitted mass (conveniently expressed in units of persons µg/m³ year/kg). The product of ΔI_i with the volume V of inhaled air per time and person yields the dimensionless exposure efficiency EE_i, which represents the fraction of the emitted pollutant mass inhaled by the population (Crettaz 2000). The exposure efficiency can be calculated by assuming a stationary situation as

$$EE_i = V/Q \int_0^R r \int_0^{2\pi} \Delta c_i(r,\varphi)\, \rho_i(r,\varphi)\, d\varphi\, dr \tag{2}$$

where (r,φ) are two-dimensional polar coordinates around the site i, $\Delta c_i(r,\varphi)$ is the stationary pollutant concentration corresponding to the arbitrary time-independent emission rate Q (note that $\Delta c_i(r,\varphi)/Q$ is independent of Q), and $\rho_i(r,\varphi)$ is the population density distribution (Nigge 2000)[1]. The upper integration limit R needs to be determined in such a way that the major part of the exposure is captured. For pollutants with atmospheric residence times of up to several days, R needs to be on the order of several thousand km (European Commission 1995).

In the following, it will be suitable to split the radial integration interval into a short range [0, 100 km] and a long range [100 km, R]. The long-range contribution $EE_{i,\,far}$ only varies on a sub-continental scale. It can therefore be approximated by a constant value for all emission sites within one country (Nigge 2000). The spatial differentiation between emissions in large cities and in rural areas is associated with the short-range contribution $EE_{i,\,near}$, which will be considered in more detail in the following.

For a given emission height, $\Delta c_i(r,\varphi)$ depends on the meteorological conditions, and in particular on the frequency distribution of wind directions at the emission site. These wind roses vary on a scale of about 10 km due to local topographical conditions (Zenger 1998). Site-specific meteorological data are therefore very difficult to obtain. This problem can be circumvented by considering classes of emission sites rather than the individual sites. Average exposure efficiencies $\langle EE_{i,\,near} \rangle$ for these classes can be calculated without first knowing $EE_{i,\,near}$ for each individual site i. This is achieved by using a statistical argument which is illustrated in Box 1.

Box 1: Statistical argument for the calculation of $\langle EE_{i,\,near} \rangle$

> Consider the class of highly densified districts within agglomerated regions (class (I,2) in Fig. 2) as an example. This class consists of 780 municipalities which are situated very close to the 56 largest German cities (class (I,1)). For an emission within one of its individual municipalities, the wind direction therefore has a significant impact on its health impact: if the neighboring large city is downwind from the emission source (in terms of the predominant wind direction), the impact is higher than if the large city is upwind. Considering all municipalities within the class, the large city can be expected to be downwind in some cases and upwind in an equal number of other cases. In the class average impact, the effects of the wind directions at the individual municipalities therefore cancel each other out.

For its mathematical formulation and empirical verification, see (Nigge 2000)[2]. Applying the statistical argument leads to

$$\langle EE_i \rangle = V/Q \int_0^{100\,km} \langle \Delta c_i(r) \rangle \langle \rho_i(r) \rangle\, 2\pi r\, dr + \langle EE_{i,\,far} \rangle \tag{3}$$

The class average exposure efficiency $\langle EE_i \rangle$ therefore only depends on the angular average $\Delta c_i(r)$ of the pollutant concentration, which is independent of the wind rose at the emission site (Nigge 2000). This eliminates much of the site-specificity. $\Delta c_i(r)$ can be calculated on the basis of generic meteorological data, which may vary between countries or sub-continental regions, but are not site-specific. For Germany, such generic meteorological data (namely combined frequency distributions of atmospheric stability and short-term wind speeds averaged over all seasons and all times during the day) were generated on the basis of results from long-term meteorological observations (Manier 1971; Christoffer and Ulbricht-Eissing 1989). The only input variable required for that purpose is the annual mean wind speed. Three geographical zones with annual mean wind speeds in the ranges of 2-3 m/s, 3-4 m/s and 4-5 m/s can be distinguished if extreme situations on the coast or in river valleys are disregarded (Gerth and Christoffer 1994).

[1] The assumption of a stationary situation can be justified as follows: 'linear exposure-response function' means (a) that the health impact of the pollutant is a function of the exposure, i.e. the time-integral of the concentration (i.e. Haber's law applies) and (b) that this function is linear. Since the partial differential equations describing the dispersion of primary pollutants are linear in both emission rates and concentrations (Seinfeld and Pandis 1998), the arguments set forth by Heijungs (1995) show that the health impact is independent of the time pattern of the emission (for a given set of atmospheric conditions) and can be calculated for the stationary case.

[2] The statistical argument requires some modification for coastal areas. For Germany, which is considered here, coastal areas were disregarded since they only represent a small part of the country.

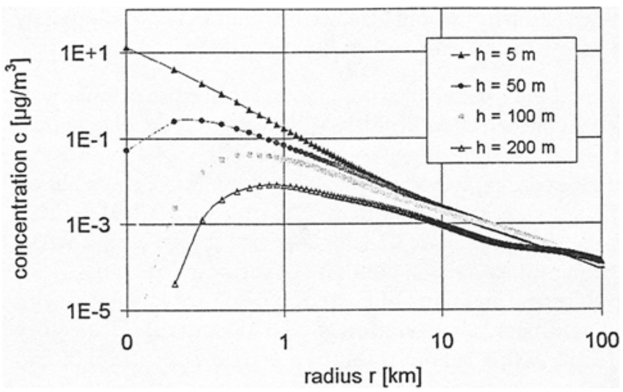

Fig. 1: Concentration of particles (PM 10) emitted at a rate of 1 kg/h at different heights

A Gaussian plume model (Janicke 1998) was used to calculate $<\Delta c_i(r)>$ from the generic meteorological data for 11 different effective emission heights h [3]. The model fulfils the requirements of the respective guideline of the German Association of Engineers (VDI 1992). **Fig. 1** shows radial concentration profiles $<\Delta c_i(r)>$ for particles (PM 10) emitted at different effective heights for the average annual mean wind speed of 3.5 m/s in Germany.

Equation (3) suggests the use of the radial population density distribution $\rho_i(r)$ within a circle of radius 100 km around the emission source as the criterion for the definition of ge-

[3] The effective emission height is the sum of the stack height and the final value of the plume rise due to mechanical impulse or thermal effects (VDI 1985). In the following, the term emission height is used in the sense of the effective emission height.

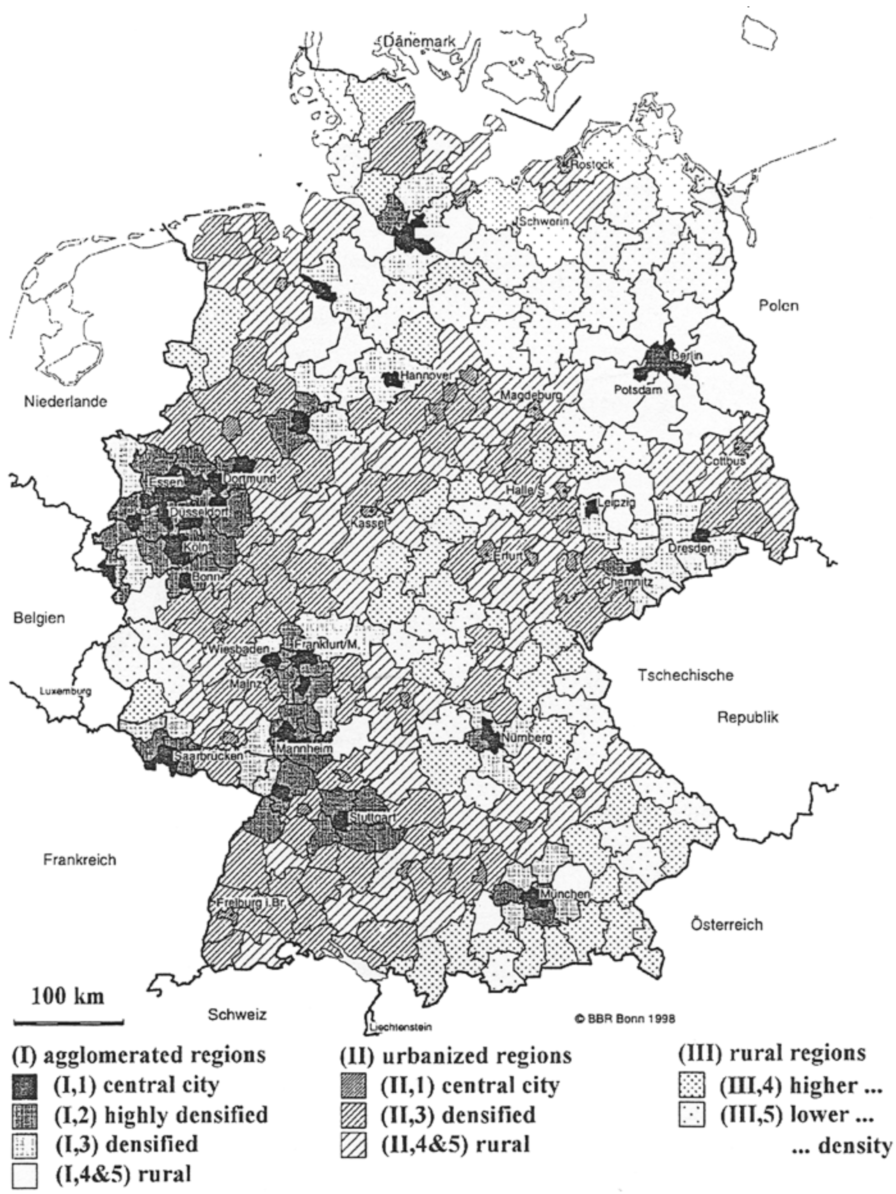

Fig. 2: Settlement structure classes in Germany (BBR 1998a). Scale 1:7,000,000

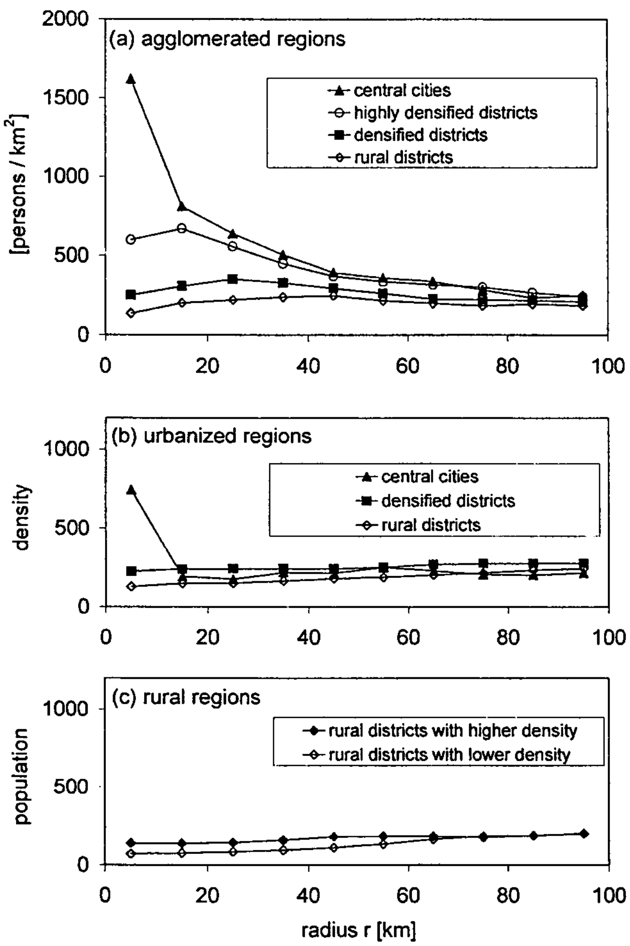

Fig. 3: Radial population density distributions for the settlement structure classes in Germany

neric spatial classes. For Germany, this criterion was operationalized on the basis of an existing official classification of settlement structures (BBR 1998a). The nine settlement structure classes range from large cities within agglomerated regions to lower density rural districts within rural regions (**Fig. 2**). Each of the classes contains between 50 and 3000 municipalities. For each municipality within a class, the radial population density $\rho_i(r)$ was calculated in increments of $\Delta r = 10$ km from data provided in (BBR 1998b) and then averaged across the class to yield $<\rho_i(r)>$ (**Fig. 3**).

The main difference between the classes is the value of $<\rho_i(r)>$ within the first 10 km (referred to as $\rho_{10\,km}$), with a spread by about a factor of 20 between the highest and lowest value. In the agglomerated regions, $<\rho_i(r)>$ also differs significantly between classes for r > 10 km, but approaches the country average population density (ρ_D = 230 persons/km^2) for $r \to 100$ km. In the urbanized regions, $<\rho_i(r)>$ is close to ρ_D throughout the range of 10 - 100 km for all classes. Increases and decreases of $<\rho_i(r)>$ in this range are due to statistical fluctuations. For rural districts in rural regions, ρ_D is approached from below, since they are far away from more densely populated areas. The fact that $<\rho_i(100\text{ km})> \approx \rho_D$ for all settlement structure classes is the reason why r = 100 km was chosen as the border between the short range and the long range, and why the long-range contribution is approximately the same for all sites within the country.

By combining these radial population density distributions with radial concentration profiles $<\Delta c_i(r)>$ such as the ones shown in Fig. 1, the short-range contribution $<EE_{i,\,near}>$ can be calculated according to equation (3). The long-range contribution $<EE_{i,\,far}>$ was calculated with the EcoSense software (IER 1998) as the mean value for 41 emission sites spread evenly across Germany. Incremental pollutant concentrations across Western, Central and parts of Eastern Europe are calculated with a Lagrangian air pollution model (Wind rose Trajectory Model) with a spatial resolution of 100 km × 100 km and combined with a database of population densities. Complete vertical mixing of the pollutants immediately after their emission is assumed, i.e. effects of emission height are neglected. The low horizontal resolution and the lack of vertical resolution of the pollutant concentration are sufficient for the long-range, but not for the short-range contribution to the exposure efficiency, which is sensitive to the emission height (see Fig. 1) and the variation of the population density on a scale of 10 km (see Fig. 3).

So far, only point sources were considered. This does not represent a limitation, however. Consider a traffic emission, for example, which can be represented as a line source. If all infinitesimal segments on the line belong to one generic spatial class, they are all characterized by the same $<\rho_i(r)>$ and the same $<\Delta c_i(r)>$. Each segment therefore contributes the same amount to the exposure efficiency $<EE_i>$. The fact that different individuals may be affected by the emissions from different segments does not matter because only the population exposure (rather than individual exposure) is considered in equation (1). The line source can therefore be condensed into a point source which emits the same pollutant mass Q per unit time. Similarly, a line source extending across n different spatial classes can be condensed into n point sources, one for each class. In that case, the share of the pollutant mass emitted within each class needs to be determined. An analogous reasoning applies to area sources.

A limitation of the method presented here is its use of a simple Gaussian plume model to calculate pollutant concentrations within the short range (up to 100 km) around the emission source. The model does not consider local climatic effects, as they may occur in cities, for example. Furthermore, the model assumes a flat terrain around the source, i.e. the effect of topographical features such as valleys or hills, or of street canyons, is not taken into account. However, the model is only used to calculate the area-integral in equation (3), rather than pollutant concentrations at individual points within that area. Errors in the calculated concentrations at individual points are therefore cancelled out to some extent, or they are insignificant, if they only concern a small fraction of the entire area. For the latter reason, the effect of the street canyon in which the emission takes place turns out to be negligible, for example. While the street canyon increases the pollutant concentration by about 2-3 times within a small area of some 10 meters in linear extension, this increased contribution to $<EE_{i,\,near}>$ is negligible

compared to the contributions from the much larger area within 100 km from the source (Nigge 2000)[4]. The extent to which <$EE_{i,\,near}$> is sensitive to built-up structures across the entire urban area (i.e. beyond the street canyon in which the emission takes place), to topographical features and to local climates, is a topic of further research.

Given the continuous nature of the spatial variation of the population densities and meteorological variables, the introduction of discrete spatial classes of emission sites is inevitably associated with a remaining variability of the impacts within the classes. A detailed statistical estimate showed this intra-class variability to be on the order of 40% of the class average impact. The averaging of the short-range exposure efficiencies over settlement structure classes and of the long-range exposure efficiencies over countries are consistent in their level of spatial detail (Nigge 2000).

2 Site-Dependent Exposure Efficiencies

2.1 Germany

For the generic spatial classes in Germany, average exposure efficiencies <EE_i> were calculated for emissions of a variety of primary pollutants (PM 2.5, PM 10, SO_2, NO_x, benzene, formaldehyde, acetaldehyde, benzo[a]pyrene, 1,3-butadiene) at different effective emission heights between 5 m and 200 m[5]. They can be directly combined with exposure-response slopes such as the ones provided by Spadaro and Rabl (1999) or Hofstetter (1998) according to equation (1). Exemplary results are presented in the following (for a complete list see Nigge 2000). An inhalation volume of 20 m^3 / (person × day) was used (Crettaz 2000).

2.1.1 Traffic Emissions

For emissions from ground-based transportation (cars, trucks, locomotives), an effective emission height of 5 m was assumed to account for a possible thermal plume rise or traffic induced turbulence. Given the fixed emission height, the focus in this section is on differences in the exposure efficiencies due to the influence of population density.

The exposure efficiencies from traffic emissions of primary pollutants with short atmospheric residence times were found to vary by about a factor of 5 between large cities in agglomerations and rural districts in Germany if the annual mean wind speed is kept at a constant value (u = 3.5 m/s). This is shown in **Fig. 4** for the example of acetaldehyde with an atmospheric residence time of 9 hours. Four out of the nine settlement structure classes are shown together with the country average, which approximately represents the five remaining classes. Due to the short atmospheric residence time of acetaldehyde (9 hours), the exposure efficiencies are dominated by the short-range contribution. The spread by a factor of 5 in the exposure efficiencies is nevertheless smaller than the corresponding spread by about a factor of 20 in $\rho_{10\,km}$ (see Fig. 3). This is because EE_{near} is dominated by contributions from the first 10 km only for high values of $\rho_{10\,km}$ (urban sites), while contributions from the range of 10 - 100 km dominate for low values of $\rho_{10\,km}$ (average and rural sites).

For substances with longer atmospheric residence times, the long-range contribution becomes more significant. This is shown for diesel particles (PM 2.5) with an atmospheric residence time of about 5 days in **Fig. 5**. Since the long-range contribution has the same value for all emission sites within one country, the relative spread between the settlement structure classes reduces to a factor of 2.2. At the same time, the absolute exposure efficiencies are increased for all settlement structure classes.

The effect of a variation of the annual mean wind speed on the short-range exposure efficiencies is shown in **Fig. 6** for the example of acetaldehyde. The country average exposure

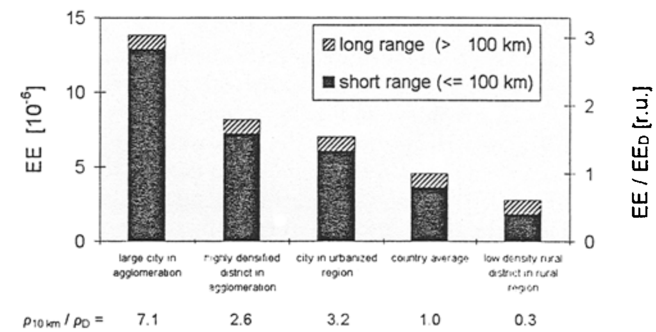

Fig. 4: Spatial variation of exposure efficiencies EE from traffic emissions of acetaldehyde in Germany. EE_D country average, $\rho_{10\,km}$ population density within the first 10 km, ρ_D country average population density

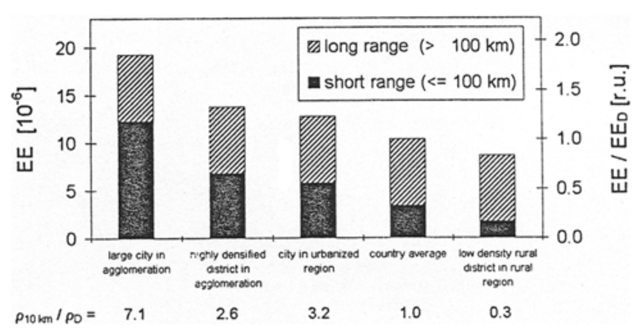

Fig. 5: Spatial variation of exposure efficiencies EE from traffic emissions of diesel particles (PM 2.5) in Germany. EE_D country average, $\rho_{10\,km}$ population density within the first 10 km, ρ_D country average population density

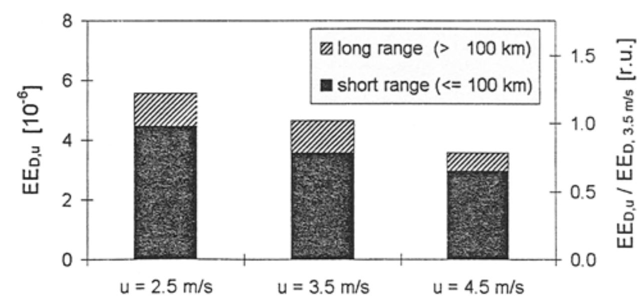

Fig. 6: Influence of the annual mean wind speed u on the country average exposure efficiency $EE_{D,u}$ from traffic emissions of acetaldehyde in Germany

[4] This refers to the effect of the street canyons on physical pollutant dispersion. Possible chemical effects of high background concentrations in street canyons on pollutant chemistry (e.g. of NO_x) were not considered.

[5] A significant share of the total health impact of formaldehyde and benzo[a]pyrene is due to the uptake of food and drinking water (Hofstetter 1998), which is not considered here.

efficiencies vary by about a factor of 1.7 across the range of typical annual mean wind speeds in Germany[6]. In cases where the short-range contribution dominates the total exposure efficiency (i.e. for substances with short atmospheric residence times, and for large cities in agglomerations even in the case of longer residence times), the annual mean wind speed therefore affects the total impacts to some extent. Its influence is nevertheless significantly smaller than that of the population density. Whether or not it is worthwhile to consider the influence of the annual mean wind speed can therefore be decided case by case, bearing in mind that this requires more detailed knowledge of the location of the emission source than a differentiation on the basis of the settlement structure classes alone.

2.1.2 Influence of Emission Height

The differentiation of exposure efficiencies between settlement structure classes decreases significantly with increasing emission height. This is shown in **Fig. 7** for emissions of particles (PM 10). The underlying effect is the decrease of the ground-level concentration within the first 10 km around the emission site (Fig. 1). The urban settlement structure classes (top 3 lines in Fig. 7) are most strongly affected by this decrease in concentration due to their high values of $\rho_{10\ km}$. Exposure efficiencies for the other settlement structure classes (average and rural) are largely insensitive to emission height. The effect of the emission height h on the exposure efficiency, hence, depends on the settlement structure class. In the example of Fig. 7, a reduction of h by a factor of 20 (from 200 m to 10 m) increases EE by a factor of 2.2 for large cities in agglomerations, but only by a factor of 1.1 for low-density rural districts in rural regions. Vice versa, the influence of the settlement structure class (approximately represented by $\rho_{10\ km}$) on EE depends on h, but is generally stronger: An increase of $\rho_{10\ km}$ by about a factor of 20 increases EE by a factor of 3.6 for h = 10m, but only by a factor of 1.7 for h = 200 m.

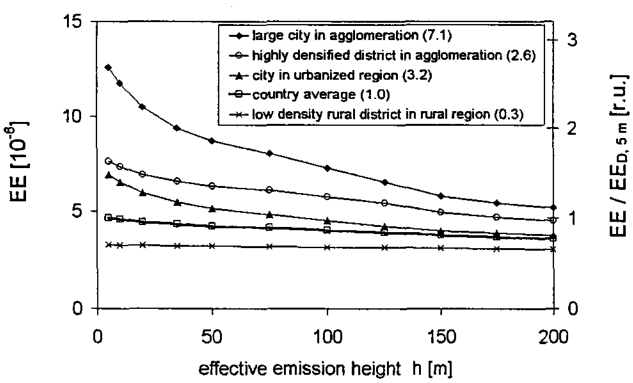

Fig. 7: Influence of effective emission height and settlement structure class on the exposure efficiency EE from emissions of particles (PM 10) in Germany. Relative population density $\rho_{10\ km} / \rho_D$ indicated in brackets for each settlement structure class. $EE_{D,\ 5\ m}$ country average for h = 5 m

[6] The long-range contributions in Fig. 6 are averages across the zones of Germany to which the respective wind speeds apply. Their variation is geographic in nature rather than directly related to wind speed.

Since five out of the nine settlement structure classes are always associated with an exposure efficiency very close to the country average, they can be merged into one generic spatial class. The spatial differentiation of the exposure efficiencies within Germany can therefore be captured by five generic spatial classes. Fig. 7 shows that the number of generic spatial classes can be further reduced for *specific* emission heights, e.g. to four classes for traffic emissions. For pragmatic purposes, it furthermore appears suitable to consider three typical effective emission heights: low (h = 5 m, in particular traffic emissions), medium (h = 50 m) and high (h = 200 m).

2.2 Europe

The definition and operationalization of generic spatial classes to capture the spatial differentiation of health impacts of primary airborne pollutants with linear exposure response functions was demonstrated for the case of Germany. The extension of the generic spatial classes to other countries is a topic of further research. However, in order to obtain a first impression of the degree of spatial differentiation to be expected for Western and Central Europe, the average exposure efficiencies, $EE_{country}$, for countries other than Germany were estimated. The Mediterranean Sea and the part of the Atlantic close to Europe were also considered. The short-range contribution was approximately determined as

$$EE_{country,\ near} = EE_{D,\ near} \times (\rho_{country}/\rho_D) \times (u_D/u_{country}) \qquad (3)$$

with

- $\rho_{country}$ average population density for the respective country (= 0 for emissions at sea)
- ρ_D average population density for Germany (230 persons/km², BBR 1998a)
- u_D annual mean wind speed for Germany (3.5 m/s, Gerth and Christoffer 1994)
- $u_{country}$ annual mean wind speed for the respective country, determined as a mean value of the measurement stations for the country listed in the European Wind Atlas (Troen and Petersen 1989).

The long-range contribution $EE_{country,\ far}$ was calculated with the EcoSense software (IER 1998) as an average over several emission sites spread evenly over the area of the country. Overall, 29 emission sites in Western and Central Europe and 10 sites in the Mediterranean and the Atlantic were considered. For countries outside of the model area, which are relevant for the case study presented in Part II (Russia, OPEC countries), $EE_{country,\ far}$ was calculated as the product of a European fate factor determined as an average for the 29 emission sites within the model area, an estimated effective population density (9 persons / km² for Russia, 33 persons / km² as an average for the OPEC countries) and the inhalation volume V.

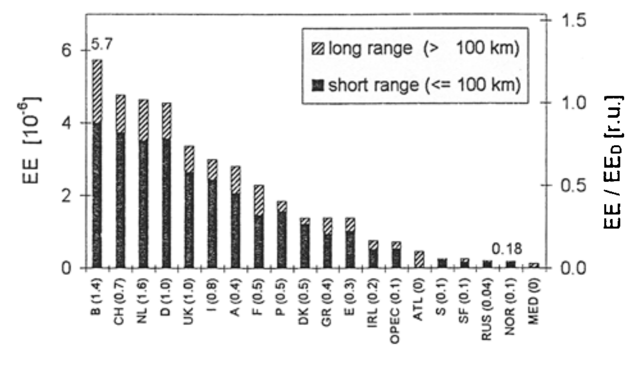

Fig. 8: Spatial variation of country average exposure efficiencies EE from traffic emissions of acetaldehyde across Europe (ATL Atlantic, MED Mediterranean). EE_D country average for Germany. Relative population density $\rho_{country} / \rho_D$ indicated in brackets for each country

Fig. 8 shows that the country average exposure efficiencies $EE_{country}$ for traffic emissions of acetaldehyde vary by about a factor of $5.7/0.18 \approx 30$. Most of the variation is due to $\rho_{country}$, while the influence of $EE_{country, far}$ and $u_{country}$ is smaller. For diesel particles, the spread reduces to about a factor of 5 due to their longer atmospheric residence time[7].

The country averages are likely to be similar to the values for emissions in rural areas of the respective countries (see Fig. 7). The high exposure efficiencies for traffic emissions in large cities in any country, on the other hand, can be expected to be similar to those for Germany, i.e. on the order of about 3 EE_D = 13.8 E-6 (Fig. 4). This is because they are largely determined by $\rho_{10\,km}$ which is similar for large cities in all countries. The overall spread between the highest and the lowest class average exposure efficiencies across Europe can therefore be expected to be in the order of a factor of $13.8/0.18 \approx 75$ for primary pollutants with short atmospheric residence times such as acetaldehyde. An analogous estimate for diesel particles yields a lower spread by a factor of 8. Differentiations of this order can actually be expected to occur within individual countries that are scarcely populated on average, but include large cities, such as within the Scandinavian countries. Generally speaking, the spread of the impact of traffic emissions between urban and rural areas within one country can be expected to be the higher the lower the average population density of the country is.

2.3 Comparison with other Studies

Both the absolute levels of exposure efficiencies as well as their relative spatial differentiation are comparable with other studies from the literature. Potting (2000) found country average exposure efficiencies for emissions of hydrogen chloride at h = 25 m across Europe to vary between 0.13 E-6 (Finland) and 2.3 E-6 (Netherlands). Average exposure efficiencies for emissions of acetaldehyde (which has a similar atmospheric residence time) at h = 5 m in these countries were found to be about twice as high here (Finland: 0.27 E-6, Netherlands: 4.6 E-6, Fig. 8), with a similar ratio between them. The differences in the absolute values are likely due to the different dispersion models used for the short range. The range of exposure efficiencies for PM10 emissions in Germany between 3.1 E-6 and 12.6 E-6 shown in Fig. 7 lies within a range across different emission heights and locations in France between 1.7 E-6 and 15.7 E-6 for emissions of SO_2 (similar residence time than PM10) as found by Spadaro and Rabl (1999). The larger range for France reflects its lower rural and higher maximum urban (Paris) population densities. European average exposure efficiencies of 10 E-6 (Hofstetter 1998) for PM2.5 and of 9.6 E-6 for fine particulates (Crettaz 2000) are comparable with the German average of 10.3 E-6 for the PM2.5 emissions shown in Fig 5, but higher than the European average of 6 E-6 determined in (Nigge 2000). The difference is probably due to differences in parameters or methods used for dispersion modeling.

[7] These factors also apply to other emission heights, since the country averages are not very sensitive to emission height (see Fig. 7).

3 Conclusion

This paper considered how the impact on human health of an emission of a primary airborne pollutant with a linear exposure-response function depends on the population density around the emission site and on the emission height, and how this spatial differentiation can be considered within a Life Cycle Assessment. Concerning the dependence on the emission site, two spatial scales can be distinguished. On a larger scale, variations between countries or sub-continental regions are due to their different average population densities. This sub-continental differentiation is insensitive to emission height. It can therefore be addressed by existing methods which are based on country average population densities and do not consider emission height. Across Europe, the sub-continental differentiation amounts to about a factor of 30 for pollutants with short atmospheric residence times of a few hours and about a factor of 5 for pollutants with long residence times of a few days.

On a smaller spatial scale, there are also variations between urban and rural areas within one country. These deviations from the country average can be captured within LCAs by the method of generic spatial classes presented here. The strongest deviations from country average impacts apply to emissions at low heights in urban areas. Urban traffic emissions are a prominent example in this regard. It is for these situations that the presented method promises the highest benefit compared to methods based on country average population densities alone (**Table 1**). Traffic emissions are also the area where the application of generic spatial classes saves

Table 1: Benefit of the proposed method over methods that consider national averages of population density alone[a]

	emission height		
	low (<25 m)	medium (25 - 100 m)	high (>100 m)
large cities in agglomerations	++	+	0
'normal' cities[b]	+	+ / 0	0
average and rural districts	0	0	0

[a] scale: ++ / + / 0 (high / medium / low benefit)
[b] cities in urbanized regions and highly densified districts in agglomerated regions

the most effort compared to a site-specific impact assessment, because the latter would have to consider a large number of individual sites along the path of the vehicle.

For traffic emissions in Germany, the urban-rural differentiation was found to be about a factor of 5 for pollutants with short atmospheric residence times and about a factor of 2 for pollutants with longer residence times. It can be expected to be higher in countries with lower average population densities (i.e. most European countries). The need to consider the urban-rural differentiation in addition to the sub-continental spatial differentiation within LCAs therefore increases with decreasing country average population density. Detailed calculations of site-dependent exposure efficiencies for generic spatial classes in countries other than Germany are a topic of further research.

Three main limitations apply to the method of generic spatial classes presented here: It is based on the assumption of a linear exposure response function, which is valid for the major pollutants from energy generation and transportation. For other pollutants, further research needs to clarify to what extent spatial differentiation of impacts also arises from variations in their background concentration. Further research is also required regarding the extent to which population exposures are sensitive to built-up structures, topographical features and local climates. The Gaussian plume model used here does not consider such effects.

Inherent in the approach of generic spatial classes is a remaining intra-class variability of impacts, which was found to be on the order of 40%. If more precise population exposures are required to reach an overall conclusion in any particular application, a more detailed, site-specific assessment needs to be carried out. In such cases, the benefit of the method presented here lies in identifying the emission sources for which a site-specific assessment is worth the effort.

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References

BBR (1998a): Aktuelle Daten zur Entwicklung der Städte, Kreise und Gemeinden. Berichte des BBR, Band 1. Bundesamt für Bauwesen und Raumordnung (BBR), Bonn

BBR (1998b): Personal communication. Bundesamt für Bauwesen und Raumordnung (BBR), Bonn

Christoffer J, Ulbricht-Eissing M (1989): Die bodennahen Windverhältnisse in der Bundesrepublik Deutschland. 2., vollständig neu bearbeitete Auflage. Berichte des Deutschen Wetterdienstes Nr. 147. Deutscher Wetterdienst, Offenbach

Crettaz P (2000): From Toxic Releases to Damages on Human Health: A Method for Life Cycle Impact Assessment, with a Case Study on Domestic Rain Water Use. Dissertation, Swiss Federal Institute of Technology, Lausanne

European Commission, Directorate-General XII (1995): ExternE, Externalities of Energy, Vols. 1-5. Prepared by ETSU, UK and IER, D. Office for Official Publications of the European Communities, Luxembourg

Gerth WP, Christoffer J (1994): Windkarten von Deutschland. Meteorologische Zeitschrift, Neue Folge, 3, 67-77

Habersatter K (1990): Oekobilanz von Packstoffen Stand 1990. Schriftenreihe Umwelt Nr. 132. Bundesamt für Umwelt, Wald und Landschaft (BUWAL), Bern

Heijungs R et al. (1992): Environmental Life Cycle Assessment of Products. Guide – October 1992. Centre of Environmental Science (CML), Leiden

Heijungs R (1995): Harmonization of Methods for Impact Assessment. Environ Sci & Pollut Res 2 (4) 217-224

Hofstetter P (1998): Perspectives in Life Cycle Impact Assessment. A Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere. Kluwer Academic Publishers, Boston

IER (1998): EcoSense software, version 2.0. Institut für Energiewirtschaft und Rationelle Energieanwendung (IER), Universität Stuttgart

Janicke L (1998): personal communication

Jolliet O, Crettaz P (1997): Critical Surface-Time 95. A Life Cycle Impact Assessment Methodology including Fate and Exposure. Swiss Federal Institute of Technology, Lausanne

Krewitt W et al. (1998): Application of the Impact Pathway Analysis in the Context of LCA. Int J LCA 3 (2) 86-94

Manier G (1971): Untersuchungen über meteorologische Einflüsse auf die Ausbreitung von Schadgasen. Berichte des Deutschen Wetterdienstes Nr. 124. Deutscher Wetterdienst, Offenbach

Moriguchi Y, Terazono A (2000): A Simplified Model for Spatially Differentiated Impact Assessment of Air Emissions. Int J LCA 5 (5) 281-286

Nigge K-M (2000): Life Cycle Assessment of Natural Gas Vehicles. Development and Application of Site-Dependent Impact Indicators. Springer-Verlag, Berlin

Potting J, Hauschild M (1997): Predicted Environmental Impact and Expected Occurrence of Actual Environmental Impact. Part II: Spatial Differentiation in Life-Cycle Assessment via the Site-Dependent Characterisation of Environmental Impact from Emissions. Int J LCA 2 (4) 209-216

Potting J (2000): Spatial Differentiation in Life Cycle Impact Assessment. A Framework, and Site-Dependent Factors to Assess Acidification and Human Exposure. Dissertation, University of Utrecht (ISBN 90-393-2326-7)

Seinfeld JH, Pandis SN (1998): Atmospheric Chemistry and Physics. From Air Pollution to Climate Change. John Wiley & Sons, New York

Spadaro JV, Rabl A (1999): Estimates of Real Damage from Air Pollution: Site Dependence and Simple Impact Indices for LCA. Int J LCA 4 (4) 229-243

Troen I, Petersen EL (1989): European Wind Atlas. Risø National Laboratory, Roskilde, Denmark

Udo de Haes HA et al. (1999): Best Available Practice Regarding Impact Categories and Category Indicators in Life Cycle Impact Assessment. Background Document for the Second Working Group on Life Cycle Impact Assessment of SETAC-Europe (WIA-2). Int J LCA 4 (2) 66-73 and 4 (3) 168-174

VDI, Verein Deutscher Ingenieure (1985): Guideline 3782, Part 3, Dispersion of Air Pollutants in the Atmosphere - Determination of Plume Rise. VDI-Verlag, Düsseldorf

VDI, Verein Deutscher Ingenieure (1992): Guideline 3782, Part 1, Dispersion of Pollutants in the Atmosphere – Gaussian Dispersion Model for Air Quality Management. Beuth Verlag, Berlin

Zenger A (1998): Atmosphärische Ausbreitungsmodellierung. Grundlagen und Praxis. Springer-Verlag, Berlin

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